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Quantifying *net* water consumption of Norwegian hydropower reservoirs and related aquatic biodiversity impacts in Life Cycle Assessment

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ABSTRACT

Compared to conventional energy technologies, hydropower has the lowest carbon emissions per kWh. Therefore, hydropower electricity production can contribute to combat climate change challenges. However, hydropower electricity production may at the same time contribute to environmental impacts and has been characterized as a large water consumer with impacts on aquatic biodiversity. Life Cycle Assessment is not yet able to assess the biodiversity impact of water consumption from hydropower electricity production on a global scale. The first step to assess these biodiversity impacts in Life Cycle Assessment is to quantify the water consumption per kWh energy produced. We calculated catchment-specific net water consumption values for Norway ranging between 0 and 0.012 m^3/kWh. Further, we developed the first characterization factors for quantifying the aquatic biodiversity impacts of water consumption in a post-glaciated region. We apply our approach to quantify the biodiversity impact per kWh Norwegian hydropower electricity. Our results vary over six orders of magnitude and highlight the importance of a spatial explicit approach. This study contributes to assessing the biodiversity impacts of water consumption globally in Life Cycle Assessment.

1. Introduction

Hydropower electricity production has the lowest carbon emissions per kWh of all conventional energy technologies (Barros et al., 2011) and can provide access to affordable and reliable energy (United Nations, 2015; Edenhofer et al., 2011; Hertwich et al., 2016). Therefore, hydropower electricity production can contribute to fulfilling two of the 17 Sustainable Development Goals (SDG), developed by the United Nations for a transition into a sustainable world (United Nations, 2015), namely SDG 7 (Affordable and Clean Energy) and SDG 13 (Climate action). However, both the United Nations Environment Program (UN Environment) (Hertwich et al., 2016) and the Intergovernmental Panel on Climate Change (IPCC) (Edenhofer et al., 2011) point out that there are potential ecological trade-offs related to hydropower electricity. Freshwater habitat alteration, land use change and water quality degradation have been identified as the main cause-effect pathways of hydropower electricity production on biodiversity (Gracey and Verones, 2016). These 3 cause-effect pathways may lead to local species extinctions (McAllister et al., 2001) of, for example, fish and macroinvertebrate species (Poff and Zimmerman, 2010; Crook et al., 2015), as well as terrestrial flora and fauna (Jansson et al., 2000; Alho, 2011; Kitzes and Shirley, 2016; Zhang et al., 2009; Tefera and Sterk, 2008). As the 17 SDGs can be viewed as a network (Le Blanc, 2015), with interdependent goals (Nilsson et al., 2016), the terrestrial and aquatic biodiversity impacts of hydropower electricity production therefore may interfere with SDG 6 (Clean Water and Sanitation) and SDG 15 (Life on Land). Thus, a sustainable hydropower development, with minimized trade offs between the SDGs (Nilsson et al., 2016; Bhaduri et al., 2016), requires an assessment of all relevant biodiversity impacts.

The report from UN Environment on green energy choices (Hertwich et al., 2016) recommends using Life Cycle Assessment (LCA) to assess potential trade-offs between renewable energy sources. LCA is a tool which is commonly used for analyzing the environmental impacts of a product or process throughout its life cycle (ISO, 2006a, b). However, the report from UN Environment does not quantify relevant biodiversity impacts from hydropower production in LCA, due to a lack of mature assessment methods (Hertwich et al., 2016; Gracey and Verones, 2016; Winter et al., 2017).

Our study focuses on freshwater habitat alteration, one of the main threats for aquatic biodiversity (Vörösmarty et al., 2010). Besides, the conservation of aquatic biodiversity has been identified as one of the key parameters for sustainable development (United Nations, 2015; Secretariat of the Convention on Biological Diversity, 2014). For

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freshwater habitat alteration, storage and pumped storage hydropower plants are most relevant, since they store water in reservoirs to allow flexible electricity production (Egré and Milewski, 2002).

The operation of hydropower reservoirs replaces various habitat types like forest, peatlands and water bodies with one large water surface (Strachan et al., 2016). This new water surface will evaporate water permanently during ice-free periods, while the possible inundated terrestrial surface will evaporate water only temporarily (Strachan et al., 2016). Due to this increased evaporation (Strachan et al., 2016), hydropower electricity production has been characterized as a large consumer of water (Mekonnen and Hoekstra, 2012). Following ISO 14046 (ISO, 2014) the alteration in evaporation caused by land use change of hydropower reservoirs is considered as water consumption. We use "water consumption" in this sense throughout the paper.

In LCAs of hydropower electricity production, a prerequisite for quantifying biodiversity impacts of water consumption is to quantify the water consumption per kWh energy produced for the Life Cycle Inventory (LCI) (ISO, 2014; Rebitzer et al., 2004; Flury and Frischknecht, 2012). This has to be done in a spatially explicit way, because underlying environmental parameters (such as precipitation, topography and climatic conditions (Flury and Frischknecht, 2012)) may vary considerably (Bakken et al., 2013; Deemer et al., 2016; Mutel and Hellweg, 2009; Mutel et al., 2012). Global assessments of water consumption values from hydropower reservoirs are not available (Bakken et al., 2016a), and in LCI databases (e.g. (Wernet et al., 2016)) spatially-explicit water consumption parameters related to hydropower reservoirs are only available for Switzerland and Brazil (Flury and Frischknecht, 2012). In addition, the dominant approach for published estimates of water consumption is the *gross* method (Bakken et al., 2013). Compared to the *net* method, the *gross* method does not account for evaporation losses of the natural lake prior to the inundation of the reservoir (Flury and Frischknecht, 2012; Scherer and Pfister, 2016a, b). As a consequence, all currently available hydropower LCI water consumption parameters represent overestimated values. Using these values leads to an overestimation of the total environmental impact. Hence, the *net* water consumption method should be the preferred approach (Bakken et al., 2013).

Water consumption leads to a reduction of the yearly average discharge downstream of the hydropower reservoir (Kumar et al., 2011; Biemans et al., 2011). Further, reservoirs can be used to store water in times of surplus and to produce electricity with a release of water during peak energy demand or drier season. This reservoir operation can in addition change the frequency of the flow magnitude (Richter et al., 1997) downstream of the hydropower reservoir (Kumar et al., 2011). However, this represents water use (ISO, 2014) (not water consumption) and is beyond the scope of this paper.

To quantify biodiversity impacts of water consumption in Life Cycle Impact Assessment (LCIA), characterization factors (CFs) quantifying the Potentially Disappeared Fraction of Species (PDF) per unit of water consumed are required (Rebitzer et al., 2004; Milà et al., 2008; Pennington et al., 2004). PDF is the recommended endpoint from the Life Cycle Initiative hosted by UN Environment to assess ecosystem quality damages (Verones et al., 2017). The existing CFs do not differentiate between the cause of water consumption. They assume that water consumption due to evaporation, water withdrawal for irrigation, industrial production, or residential needs, has in principle the same impact on freshwater biodiversity (Tendall et al., 2014; Hanafiah et al., 2011).

Spatially-explicit CFs for water consumption impacts on aquatic biodiversity have been globally developed for areas below 42°N, and for Europe with a focus on Switzerland (Tendall et al., 2014; Hanafiah et al., 2011). All these CFs are based on Species-discharge relationships (SDR), which relates the discharge rates of given rivers to the associated species richness (Xenopoulos and Lodge, 2006).

The main reason for excluding areas at latitudes above 42°N is that these river basins were recently (in geological time) glaciated and have not had time to reach their maximum species richness potential (Tendall et al., 2014; Hanafiah et al., 2011). This means that for Canada, Norway, Sweden, Finland, and Iceland, which have been glaciated during the last glacial maximum (Clark et al., 2009) and accounted together for 11.8% of the global hydropower electricity production in 2016 (IEA, 2017), no spatially-explicit CFs exist to assess impacts of water consumption on biodiversity.

The first aim of this study is to calculate *net* water consumption values of hydropower electricity production for the LCI. Due to data availability we limit the calculation of *net* water consumption values to Norway, which is one of the top-ten hydropower electricity producers worldwide (Manzano-Agugliaro et al., 2013) and where the government corroborates that hydropower electricity production has significant environmental impacts on rivers that should be assessed (Norwegian Government Ministries and Offices, Meld. St. 25, 2015–2016). However, our suggested framework has the potential to be used in other regions, given that data are available.

The second aim of the study is to develop the first spatially-explicit CFs for water consumption in post-glaciated regions, based on regionally specific SDRs for fish, accounting for local variation in fish fauna by delineating regions with the same postglacial freshwater fish immigration history. Due to data availability, we only develop CFs for Norway. The output is a set of catchment specific CFs that express the fish biodiversity loss in PDF per unit water consumed for Norway. Due to data availability and the complexity to reconstruct the postglacial immigration history of species, we only consider fish species in this study, as they are good indicators of ecosystem health (Schiemer, 2000).

The third aim of this study is to use the provided LCI values and CFs to calculate the impact on aquatic biodiversity of water consumption from Norwegian hydropower reservoirs in LCA. Further, it enhances the development of CFs quantifying the impact on aquatic biodiversity of water consumption in other glaciated regions.

2. Method

2.1. Quantifying water consumption for the Life Cycle Inventory

Water consumption can be divided into three components: green water consumption (consumptive use of rain water), blue water consumption (consumptive use of ground or surface water) and grey water consumption (the volume of water polluted) (Mekonnen and Hoekstra, 2012). The water consumption quantified in this study follows the ISO 14046 (ISO, 2014) and only concerns blue water consumption in the form of evaporation from reservoirs during the use phase for storage hydropower plants (ISO, 2014).

Two main methods exist to calculate water consumption from hydropower reservoirs: *gross* water consumption and *net* water consumption. *Gross* water consumption is the most commonly used method and equates the evaporation of the actual reservoir divided by the annual electricity production. As the reservoir area could originally have been either a natural lake or a terrestrial area the *gross* water consumption does not account for evaporation losses *prior to* the construction of the hydropower reservoir. This is leading to an overestimation of the water consumption (Bakken et al., 2013). In contrast, the *net* water consumption method accounts for the evaporation losses *prior to* the construction of the hydropower reservoir, i.e. the evaporation rates from the actual reservoir surface area minus the evaporation rates *prior* to the reservoir construction divided by annual power production. Because the majority of Norwegian hydropower reservoirs are dammed natural lakes (Dorber et al., 2018), the net water consumption is used in this study. Consequently, calculation of the *net* water consumption requires open water evaporation rates from the actual reservoir surface, as well as land use change information, including evaporations rates of the terrestrial area *prior* to reservoir

inundation. To estimate open water evaporation, several methods, including empirical, water budget, energy budget, or mass transfer exits. These methods can all be applied either alone or in combination (Mekonnen and Hoekstra, 2012; Mengistu and Savage, 2010). The Penman-Monteith equation with heat storage, a combination method of energy budget and mass transfer, is often considered most suitable for estimating open water evaporation from hydropower reservoirs (Mekonnen and Hoekstra, 2012; Rosenberry et al., 1993). However, this approach can neither be applied to Norway nor globally, as the necessary in situ data on, for example, water temperature and wind speed, are not available in the required, detailed spatial scale (Finch, 2001). Therefore, we use the potential evapotranspiration (PET) as proxy for the open water evaporation (Lee et al., 2014), as for example done by Pfister et al. (2011) and Scherer and Pfister (2016a). Evapotranspiration (ET) can be defined as the amount of water transferred to the atmosphere by evaporating water from plants or soil surfaces (Mu et al., 2007). PET is the amount of evapotranspiration which occurs when an infinite amount of water is available (Westerhoff, 2015). AET is defined as the amount of evapotranspiration happening under local water conditions (Westerhoff, 2015), affected by annual rainfall, vegetation type and climatic conditions (Zhang et al., 2001).

The validity of this assumptions is confirmed by, e.g., Lee et al. (2014) who report a difference of 5% between satellite based PET estimates and open water evaporation measurements, and Douglas et al. (2009) who report a difference of up to 6% between Penman–Monteith PET estimates and open water evaporation measurements. However, the rates can differ depending on the PET estimation method (Rosenberry et al., 1993; Douglas et al., 2009; Lu et al., 2005).

The evaporation rates from the actual reservoir equals the potential evapotranspiration of the actual reservoir surface area. To calculate the evaporation rate *prior* to the reservoir construction, land use information *prior* to reservoir construction is needed. The evaporation rate *prior* to the reservoir construction is the PET occurring on the natural water surface area plus the AET occurring on the later inundated terrestrial land area. As there is no change in PET from changing $1\,m^2$ natural water surface area to $1\,m^2$ reservoir surface area, the net water consumption only considers the difference between PET and AET of the inundated land area. As the water consumption of all hydropower reservoirs in a catchment leads to a discharge reduction in the same main river, the catchment level is chosen as a system boundary. Thus, the *net* water consumption [m^3/kWh] in catchment x for the LCI can be calculated according to Eq. (1).

$$\text{Net water consumption}_x = \frac{\sum_{y=0}^{k} \frac{((\text{PET}_y - \text{AET}_y) \times \text{ILA}_y)}{1000}}{\sum_{y=0}^{k} \text{ER}_y} \quad (1)$$

where k is the number of reservoirs with inundated land data in catchment x, *PET* is the average yearly potential evapotranspiration in mm/year of reservoir y, *AET* is the average actual evapotranspiration in mm/year of reservoir y, *ILA* is inundated land area in m^2 due to the reservoir creation of reservoir y and *ER* is the average annual electricity production in kWh of reservoir y.

The average yearly potential evapotranspiration and average yearly actual evapotranspiration were obtained from the MODIS Global Evapotranspiration Project (MOD16) (Mu et al., 2007, 2011; University of Montana, 2011). MOD16 is based on the Penman-Monteith equation and by using Land Cover Data, the Leaf Area Index and a modified version of the Normalized Difference Vegetation Index, the MOD16 is able to distinguish the evaporation rates of different vegetation types. It offers an average potential evapotranspiration and average actual evapotranspiration for the period 2000–2013 in a 1-km^2 resolution for the whole globe (Mu et al., 2011).

To calculate *PET* we averaged the MOD16 PET values inside the actual reservoir surface area at highest regulated water level (RSA) provided by the Norwegian Water Resources and Energy Directorate (NVE) (NVE (The Norwegian Water Resources and Energy Directorate), 2016) (see Supporting Information 2 (SI2)). *AET* could not be calculated directly, because MOD16 assesses the status after reservoir inundation, and information about the vegetation and soil composition prior to inundation does not exist (Dorber et al., 2018). Therefore, we had to assume that a buffer around the shoreline of the actual reservoir, represents the vegetation and soil composition prior to inundation. Based on this assumption we assessed *AET* by averaging the MOD16 actual evapotranspiration in a 2-pixel buffer around the shoreline of the actual reservoir in ArcGIS10.3 (ESRI (Environmental Systems Research Institute), 2014) (see Supporting Information 2 (SI2)). The sensitivity of this assumption will be tested and discussed in Section 3.2. Inundated land area data are obtained from Dorber et al. (2018).

2.2. Uncertainty and sensitivity of water consumption calculations

Main contributors to uncertainty of the calculated *net* water consumption are evaporation estimates, inundated land area estimates and water-level fluctuations. For evaporation estimation from the MOD16 project, Mu et al. (2011) report an average mean absolute bias of 24.6% for the *AET* value. We account for this uncertainty by calculating a *net* water consumption due to AET using 24.6% higher and lower *AET* values (see Supporting Information 1 (SI1), section S2 and SI2). To account for uncertainty related to inundated land area assessment, we calculate a *net* water consumption with the standard deviation (SD) of the adjusted inundated land area data from Dorber et al. (2018). Further, Dorber et al. (2018) calculated the inundated land area related to the actual reservoir surface area at the highest regulated water level. The common operational scheme for Norwegian reservoirs is characterized by a distinct decline in water level during winter followed by a significant increase in spring, and an almost stable water level during summer and autumn (Mjelde et al., 2012; Eloranta et al., 2018). Additionally, most Norwegian hydropower reservoirs are generally filled to less than 90% of maximum capacity (Norwegian Water Resources and Energy Directorate, 2017). Consequently, the actual reservoir surface area at the highest regulated water level may not be reached over the whole year. Thus, our *net* water consumption values, which do not cover seasonal water-level fluctuations, are most likely overestimations. As the relationship between water level and water surface area is not available for Norwegian hydropower reservoirs (Mekonnen and Hoekstra, 2012), the uncertainty of this temporal aspect cannot be quantified directly. Therefore, we test the sensitivity of water-level fluctuations on the calculated *net* water consumption value by reducing the inundated land area. To test the sensitivity of the assumption that a buffer around the actual reservoir represents the vegetation prior to inundation, we calculate the *net* water consumption in addition with a 1-pixel buffer (see SI2).

2.3. Aquatic species loss per unit change of discharge

To assign biodiversity damage to water consumption from the LCI in LCIA on a damage level, a characterization factor for each catchment needs to be developed. The CF denotes the Potentially Disappeared Fraction of Species per unit of water consumption (Verones et al., 2017). In this study, we used the Species-discharge relationship concept already applied within LCIA for the derivation of water consumption CFs (Tendall et al., 2014; Hanafiah et al., 2011). As species richness is positively correlated with mean annual discharge (Oberdoff et al., 1995; Poff et al., 2001; Xenopoulos et al., 2005), the SDR is a model that relates river discharge to species richness within a catchment (Xenopoulos and Lodge, 2006). This relationship can therefore be used to predict the species loss per unit change of discharge (Xenopoulos and Lodge, 2006).

In regions where SDRs have already been developed, fish species richness variability can be statistically explained as a function of mean annual discharge (Oberdoff et al., 1995). However, in the northern Hemisphere, including Norway, species richness variability is

additionally explained by historical glaciation events and postglacial immigration history (Oberdorff et al., 1997; Reyjol et al., 2007; Leprieur et al., 2009), which caused variation on a local scale. An SDR developed for the whole of Norway is weak, because even today postglacial immigration plays an important role for species richness variability (Hanafiah et al., 2011). Therefore, the first step in developing regional SDRs for Norway is to identify catchments with similar glaciation and dispersal history. Within each catchment, species richness is subsequently correlated with mean annual discharge. Consequently, catchment-specific SDRs are calculated.

2.3.1. Identifying catchments with similar glaciation and dispersal history

During the last glacial maximum the northern parts of Europe were covered by ice or permafrost (Reyjol et al., 2007). Many fish species in the northern part of the continent were unable to migrate along a north–south gradient and therefore became locally extinct (Reyjol et al., 2007). The surviving fish species shifted south into so-called glacial refugia (Reyjol et al., 2007; Leprieur et al., 2009; Hänfling et al., 2002; Refseth et al., 1998; García-Marín et al., 1999; Griffiths, 2006). From these refugia, recolonization of all freshwater fish species into Scandinavia occurred when the ice retreated after the last glaciation (approx. 10,000 years ago) (Refseth et al., 1998). As catchments are separated by barriers that are insurmountable for freshwater fish (land masses or oceans), the movement of freshwater fish into Norway is defined by the connectivity of water bodies through rivers and streams (Leprieur et al., 2009). Saltwater-tolerant (anadromous) fish were able to colonize coastal Norway via the sea from the West, while non-anadromous freshwater fish probably colonized Norwegian water courses from the East or Southeast from the Baltic Sea refugium, or from the south following the retreating glacial front (Refseth et al., 1998). Colonization via the seas is considered a fast process in comparison to colonization via land masses (Oberdorff et al., 1997; Reyjol et al., 2007). Fish migration via land masses could only happen during marine regressions when sea levels decreased and new freshwater connections between catchments became possible (Reyjol et al., 2007). During the last glacial maximum a decrease in sea levels by 20 m occurred (Patton et al., 2017). Alternatively, fish migration via land mass occurred when the water of melting glaciers connected catchments located on opposite sides of mountain ridges (Oberdorff et al., 1997; Reyjol et al., 2007).

To account for the colonization history in Norway via the seas, we select catchments according to their associated marine ecoregion (Spalding et al., 2007). This assumes that the distance to the refugia and also the recolonization time is equal for all catchments draining into the same marine ecoregion. Following Reyjol et al. (2007), the selection of catchments by marine ecoregions also accounts for colonization via marine regression, assuming that these catchments experienced the same sea-level lowering. To account for colonization through surface waters in land masses, we select catchments by the freshwater ecoregions they belong to. Freshwater ecoregions are partially defined by geological processes, speciation, glaciation history, climatic and physiographic patterns, and dispersal barriers, with a focus on freshwater fish species (Abell et al., 2008). Thus, a region with similar colonization history is delineated by those catchments located in the same freshwater region and draining into the same marine ecoregion (Fig. 2) (SI1, S3).

2.3.2. Developing regional SDRs for Norway

Species-discharge relationships for each of the identified regions with similar colonization history are derived by curve-fitting the relationship between the discharge rates and the fish species richness of a given catchment. Annual runoff for the period 1961–1990 in each catchment is provided by NVE (NVE (The Norwegian Water Resources and Energy Directorate), 2016). We use the oldest available period, to represent the natural flow situation before hydropower. Fish species occurrence data are obtained through the publicly available database and map services Artsdatabanken (The Norwegian Biodiversity Information Centre and GBIF Norway, 2017) and GBIF (The Norwegian Biodiversity Information Centre and GBIF Norway, 2017; Finstad et al., 2017; Hesthagen and Gravbrøt, 2017; Natural History Museum, University of Oslo, 2017; Vang, 2017a, b). We exclude freshwater fish species classified as introduced from Fishbase (Froese and Pualy, 2018) and obtained 140,311 fish occurrence points, collected between 1869 and 2017 in 1463 catchments (SI1,S4). For reasons of comparability, we use the power function commonly employed in LCA to calculate the SDR (Tendall et al., 2014). The SDR function is solved analytically, as shown in Eq. (2).

$$S = a \times x^{b}$$

$$dS = (b \times a) \times x^{(b-1)} \tag{2}$$

a and b are model coefficients produced by the regression model, whereas x signifies the discharge rate [m^3/y] of the catchment in question. The SDR equates how many species S we would expect within a catchment, whereas dS (the derivative of the SDR power function) tells us how the number of fish species changes as we change the discharge by one unit (m^3/y).

As some sites are more likely to be surveyed than others (Phillips et al., 2009), the number of species occurrence points varies in each catchment. We assume that the accuracy of species richness estimates increases when more occurrences are recorded in a catchment. To account for this assumption we weigh the power function fitting by the total number of occurrence records in each catchment (SI1, S4) (Motulsky and Christopoulos, 2004). Power function fitting was performed in MATLAB version R2015a using the nonlinear least squares method (Mathworks, 2015). We do not calculate SDRs for Norwegian catchments with rivers that flow into Sweden or Finland or catchments in Norway where more than 30% of the area is located outside Norway, because discharge and species richness data for these catchments are not available in an exhaustive and comparable way.

2.3.3. Calculation of the characterization factor

The characterization factor (CF) [PDF*y/m^3], consisting of a Fate Factor (FF) [m^3*y/m^3*y] and Effect Factor (EF) [PDF*y/m^3], quantifies the downstream impact of water consumption in catchment x on freshwater fish species in Norway, and can be expressed by Eq. (3). The FF models the river discharge reduction of a unit water consumed and the EF relates the intensity of a unit water consumed to a quantified biodiversity effect.

$$\mathrm{CF_x} = \mathrm{FF_x} \times \mathrm{EF_x} = \frac{\mathrm{dQ}}{\mathrm{dW}} \times \frac{\frac{\mathrm{dS}_x}{\mathrm{R}_x}}{\mathrm{dQ}} \tag{3}$$

The FF is adopted from Hanafiah et al. (2011), where dQ is the marginal change in discharge [m^3/y] and dW is the marginal change in water consumption [m^3/y]. The FF equals one, as one unit change in water consumption (e.g. 1 m^3 evaporation) leads to one unit reduction of river discharge. For EF, dS is the derivative of the SDR power function developed for the related region in Norway (see Eq. (3)), used to find the species loss per unit change of discharge. R is the total fish species richness of catchment x, which is the maximum number of species predicted by the SDR. The ratio of dS to R gives the potentially disappeared fraction of fish species loss per unit water consumption. In our case, dQ is always 1 m^3/y, to link it with the water consumption of the Life Cycle Inventory. We calculated the 95% simultaneous confidence intervals of the fitted power function and the related coefficients in each region with MATLAB version R2015a (Mathworks, 2015) to quantify the uncertainty of the CFs.

Water consumption due to water withdrawal for irrigation, industrial production, or residential needs can in principle have the same impact on the freshwater biodiversity (Tendall et al., 2014; Hanafiah et al., 2011). Therefore, the developed CFs are applicable to all fields of blue water consumption in Norway, with related LCI data, and are not limited to the quantification of water consumption impacts from

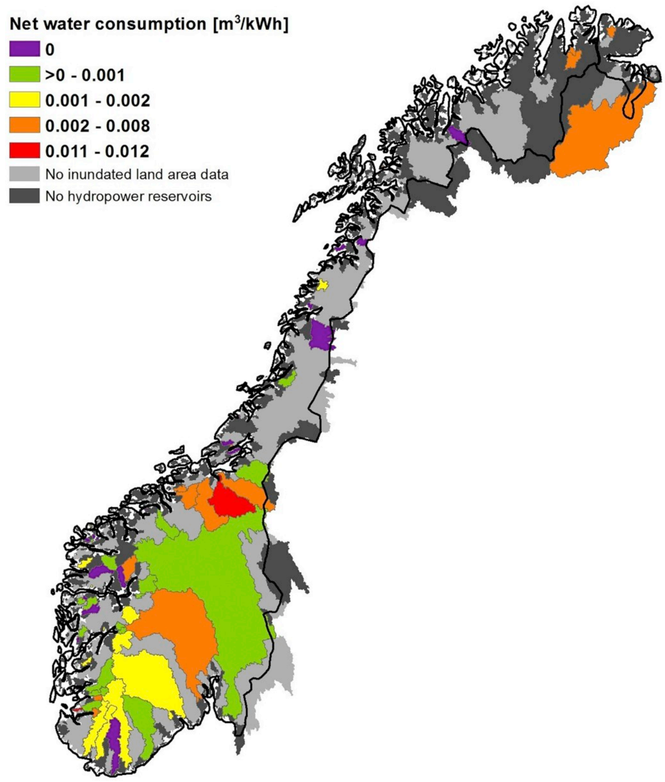


Fig. 1. *Net* water consumption per kWh calculated from the adjusted inundated land area for Norway (Thematic Mapping API World Borders Dataset, 2009). In grey areas no inundated land information was available. In the dark grey areas no hydropower reservoirs exist. Catchment information obtained from the Norwegian Water Resources and Energy Directorate (NVE (The Norwegian Water Resources and Energy Directorate), 2016).

hydropower. To showcase the applicability of our results we calculate the impact on aquatic biodiversity of water consumption from hydropower electricity production in Norwegian catchments in Section 3.5.

3. Results

3.1. Net water consumption

We calculate *net* water consumption values for 63 out of 1833 Norwegian catchments including 107 reservoirs (Fig. 1). For the remaining catchments no *net* water consumption values could be calculated, due to a lack of reservoirs with inundated land area data (Dorber et al., 2018). The average *net* water consumption was 0.0016 m^3/y, with a minimum of 0 m^3/kWh and a maximum of 0.012 m^3/kWh. A value of 0 m^3/kWh indicates that a natural lake existed prior to the dam construction and that its surface area was not increased.

3.2. Uncertainty and sensitivity of water consumption

Accounting for uncertainty in the actual evapotranspiration results in an average *net* water consumption due to AET that differs by 0.0007 m^3/kWh, respectively 42.6% relative to the average *net* water consumption presented before. Hence, the average *net* water consumption due to AET, varies between 0.0009 m^3/kWh and 0.0023 m^3/kWh. Accounting for inundated land area estimation uncertainty results in an average *net* water consumption due to inundated land area that varies between 0.0014 m^3/kWh and 0.002 m^3/kWh, respectively −20.1% and 22.9% relative to the average *net* water consumption. The calculation procedure for the inundated land area uncertainty reveals that a reduction of the inundated land area by 1% results in an average reduction of 0.000016 m^3/kWh, respectively 1% relative to the average *net* water consumption. The difference between the *net* water consumption calculated with actual evapotranspiration within a 2-pixel buffer in comparison to a 1-pixel buffer varies between 11.2% and −9.7%, with an average of 1.2%. For a visualization of the estimated uncertainty and further explanations see Supporting Information 1, Section S2.

3.3. Regional SDRs

For Norway, we identify eight regions where catchments are draining into the same marine or freshwater ecoregion (Fig. 2). We develop an SDR for five of the eight identified regions. It is not possible to develop a SDR for region 4 and region 6, because they only consist of one catchment each. Region 8 includes only catchments with rivers flowing into Sweden and Finland, so no SDR is developed, due to a lack of data. The fit of the power functions, reflected in the R^2, varies between 0.43 and 0.81.

3.4. Characterization factors

Based on the five SDRs, we calculate characterization factors for 1790 of 1833 catchments in Norway varying between $7.1*10^{-12}$ PDF*y/m^3 and $8.0*10^{-7}$ PDF*y/m^3 (Fig. 3). For the remaining 43 catchments, no characterization factors are calculated as these are either situated in region 4 and region 6 or overlapped with Sweden.

The CFs in Fig. 3 do not follow the pattern of the regions identified in Fig. 2. The new pattern can be explained by the fact that we are calculating the Potentially Disappeared Fraction of Species as the species loss per m^3 water consumed divided by the fish species richness of catchment *x*. Even if the relative species loss per m^3 water consumed is the same for a small and a large catchment, the small catchment will get the comparably higher PDF*y/m^3 value, because it has a comparably lower fish species richness. For further explanations, see Supporting Information 1, Section S6.

By using the 95% confidence intervals of the fitted power function we estimate an uncertainty of respectively ± 28% in Region 1, ± 5% in Region 2, ± 18% in Region 3, ± 8% in Region 5, and ± 9% in Region 7, relative to the characterization factors. Therefore, the CFs considering uncertainty vary between $6.00*10^{-12}$ PDF*y/m^3 and $8.35*10^{-7}$ PDF*y/m^3. The CF values are provided in Supporting Information 1, Section S6 and Supporting Information 2.

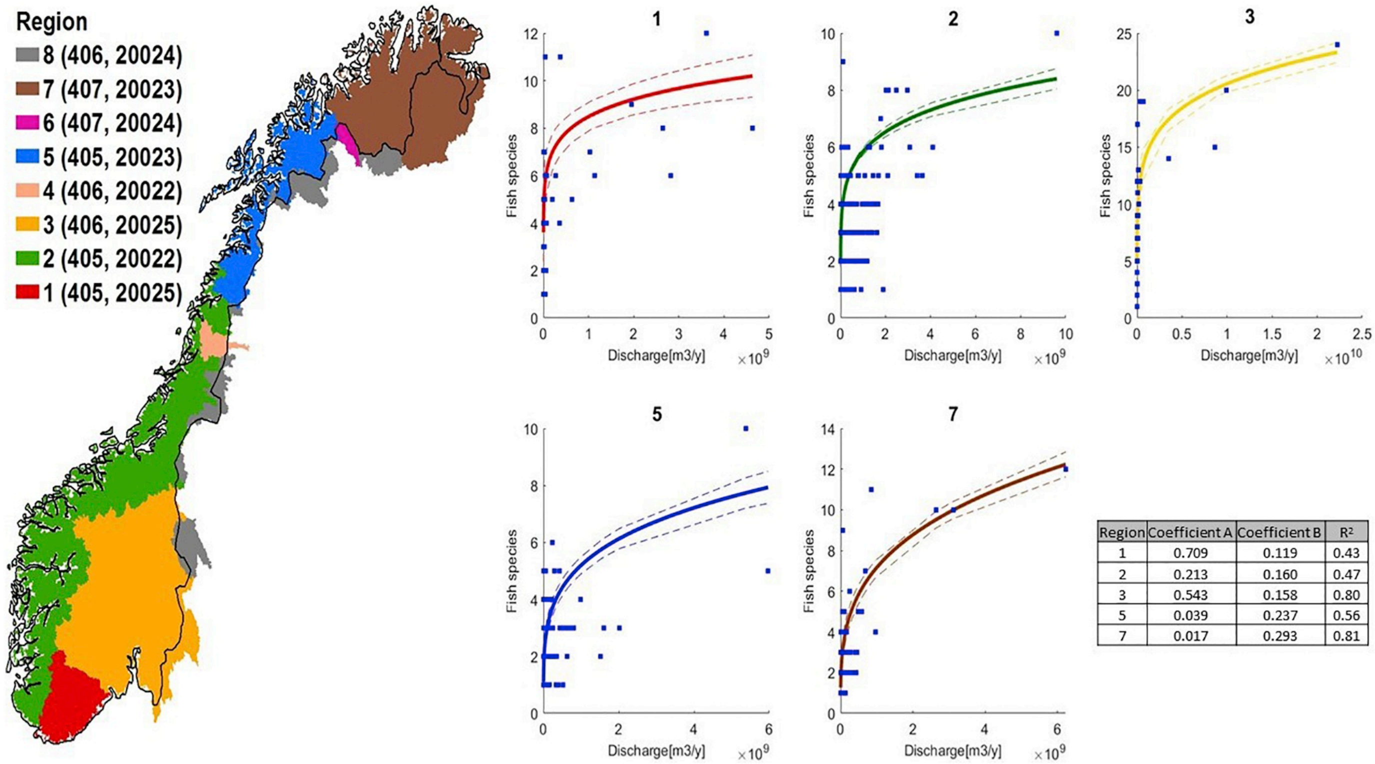
3.5. Application

To showcase the applicability of our results we calculate the impact on aquatic biodiversity of water consumption from hydropower electricity production in Norwegian catchments by multiplying the *net* water consumption LCI values with the regional CFs assessed in this study (Fig. 4). The functional unit is 1 kWh hydropower produced. In cases where no catchment-specific inventory parameter is available we average the available *net* water consumption value on freshwater ecoregions (405 = 0.0014 m^3/kwh; 406 = 0.0023 m^3/kwh; 407 = 0.0038 m^3/kwh) (Abell et al., 2008). We used freshwater ecoregions as they can be used to categorize water bodies (Abell et al., 2008). However as for this purpose no standard LCA methodology exists (Mutel et al., 2012), our approach should be seen as one example.

4. Discussion

4.1. Water consumption for the Life Cycle Inventory

This is the first study providing *net* water consumption values of storage hydropower plants for the Life Cycle Inventory with estimated uncertainty. The unit of the modelled *net* water consumption is m^3/kWh, which is in accordance with the unit of m^3 water consumption in



Region	Coefficient A	Coefficient B	R²
1	0.709	0.119	0.43
2	0.213	0.160	0.47
3	0.543	0.158	0.80
5	0.039	0.237	0.56
7	0.017	0.293	0.81

Fig. 2. Left: Regions where catchments are draining into the same marine or freshwater ecoregion (3 digit number = Freshwater Ecoregion (Abell et al., 2008) code; 5 digit number = marine ecoregion (Spalding et al., 2007) code); Right: Developed SDRs (solid line) and confidence interval (dashed lines) with corresponding coefficients and R^2 for each of the regions.

the commonly used water consumption inventory (Koellner et al., 2013). This makes our *net* water consumption values calculated for Norwegian catchments directly implementable in LCI databases (Pekel et al., 2016; Lehner et al., 2011). The average *net* water consumption for Norway in our study across all investigated catchments was 0.0016 m^3/kWh, which is 25% smaller than the existing value in the Ecoinvent database (0.002 m^3/kWh) (Flury and Frischknecht, 2012). Thus, current Life Cycle Impact Assessments of water consumption from Norwegian hydropower reservoirs would overestimate a potential impact by 25%. *This* highlights that spatially-explicit inventory modelling is needed (ISO, 2014; Flury and Frischknecht, 2012; Bakken et al., 2013; Mutel and Hellweg, 2009) to assess the impact of water consumption on a global scale in LCA (Núñez et al., 2016). By using remote sensing assessed reservoir inundated land area (Pekel et al., 2016) and global hydropower reservoirs data (Lehner et al., 2011) in combination with the global MOD16 evaporation model, the methodology for Norway developed in this study has the potential to be applied globally. Therefore, this study contributes to providing a method to assess the biodiversity impact of water consumption from hydropower electricty production, which is a requirement for LCA purposes (Núñez et al., 2016).

We choose the MOD16 model with the Penman-Monteith equation, as it provides global evaporation values. It therefore enhances the development of *net* water consumption values for the LCI of hydropower electricity production on a global scale. The basis for our calculated *net* water consumption are the evaporation values under the climatic conditions from 2000 to 2013. These values do not accommodate for the fact that evaporated water may return as precipitation in the same catchment (Bakken et al., 2013). This may lead to an overestimation of the *net* water consumption. Abstraction of water in hydropower tunnels is also not included. If evaporation rates change under further climate change scenarios (Hanafiah et al., 2011), new *net* water consumption values will have to be calculated.

A *net* water consumption value for only 63 of 1833 catchments could be calculated, due to a limited number of reservoirs with inundated land area (Dorber et al., 2018). However, the availability of data on 63 catchments, including 107 reservoirs, adds important information from Norway to the 52 reservoirs assessed to calculate a water consumption for Switzerland in the existing Ecoinvent database (Flury and Frischknecht, 2012). Seven out of the 107 reservoirs are used as multipurpose reservoirs (Dorber et al., 2018; NVE (The Norwegian Water Resources and Energy Directorate), 2016). In these cases hydropower electricity production might not be the only factor causing water consumption, wherefore the resulting water consumption in multipurpose reservoirs should be allocated to all use purposes (Bakken et al., 2016b; Scherer and Pfister, 2016b). For four out of the seven multipurpose reservoirs, a net water consumption of 0 m^3/kWh was calculated. Following, in these cases allocation would not have an influence on the results. As the remaining three reservoirs are only used as flood protection dams in addition to hydropower electricity production, we have not included an allocation factor. Consequently, our calculated *net* water consumption values may overestimate the water consumption caused by electricity production for these three hydropower reservoirs.

During the whole life cycle of a storage power plant, the dam construction and the reinvestment contribute additionally to the total water consumption. For Norway a contribution of 67.8% from the use-phase of storage power plants of the total water consumption has been reported (Bakken et al., 2016a). This is indicating that water consumption of the use-phase is the major contributor to the total water consumption.

4.2. Uncertainty and sensitivity levels in water consumption estimation

The average *net* water consumption considering AET uncertainty varies between 0.0009 m^3/kWh and 0.0023 m^3/kWh. Accounting for

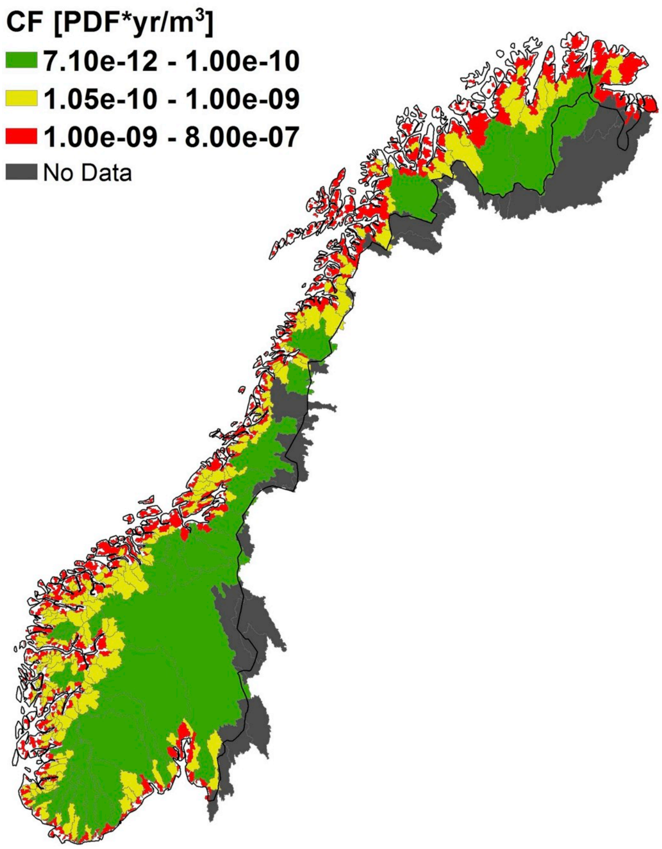


Fig. 3. Results of catchment-specific characterization factors quantifying the marginal impact of net water consumption on freshwater fish species in PDF*y/m^3. Catchment information obtained from the Norwegian Water Resources and Energy Directorate (NVE (The Norwegian Water Resources and Energy Directorate), 2016).

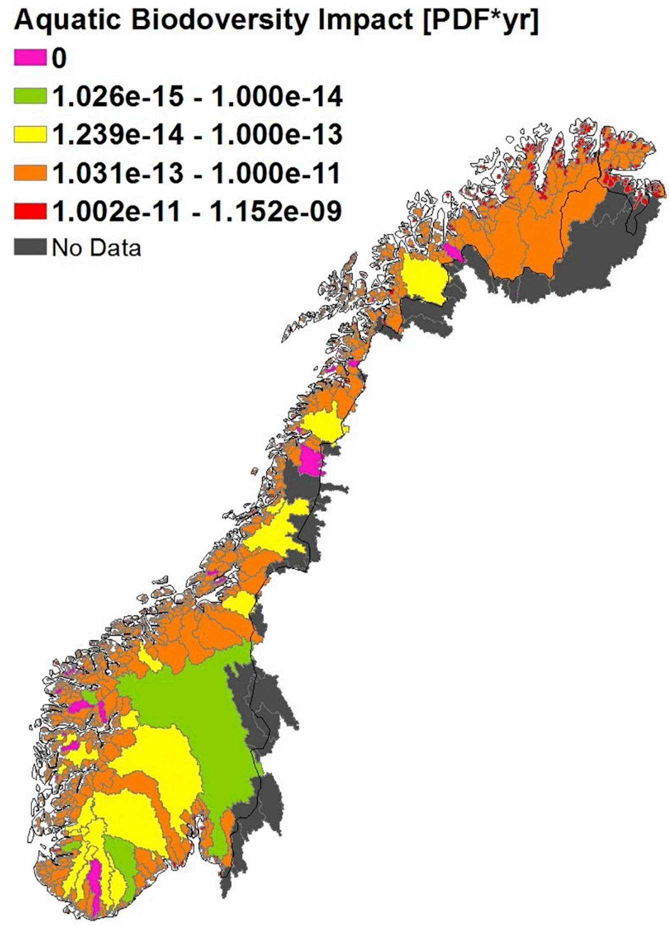


Fig. 4. Impact on aquatic biodiversity of water consumption from 1 kWh hydropower electricity production in Norwegian catchments [PDF*yr]. Catchment information obtained from the Norwegian Water Resources and Energy Directorate (NVE (The Norwegian Water Resources and Energy Directorate), 2016).

uncertainty of the inundated land area results in an average *net* water consumption that varies between 0.0014 m^3/kWh and 0.002 m^3/kWh.

We have investigated evaporation and inundated land uncertainty separately. A combined assessment of both uncertainties is not possible, as the standard deviation of the inundated land area is obtained directly for each reservoir, while the evaporation uncertainty is only available as average mean absolute error based on field stations not located in Norway. A reduction of the inundated land area by 1% results in an average reduction of 1% relative to the average *net* water consumption. This indicates a linear relationship between the calculated *net* water consumption and water-level fluctuations. However, as the relationship between water level and water surface area is not available for Norwegian hydropower reservoirs (Mekonnen and Hoekstra, 2012), the overestimation cannot be quantified directly. This highlights the need for quantifying the relationship of water level and water surface for all Norwegian hydropower reservoirs, to account for water-level fluctuations in *net* water consumption values.

The proportional difference between the *net* water consumption calculated with actual evapotranspiration within a 2-pixel buffer in comparison to a 1-pixel buffer varies between 11.2% and −9.7% with an average of 1.2% (SI2). Our finding, that the average proportional difference between the *net* water consumption calculated with actual evapotranspiration within a 2-pixel buffer compared to a 1-pixel buffer is only 1.2%, shows that vegetation and thus actual evapotranspiration is not sensitive to distance.

4.3. Regional SDRs for Norway

Our five SDRs with an R^2 between 0.43 and 0.81 lie in the range of the R^2 between 0.35 and 0.90 reported by Tendall et al. (2014) for Europe and the R^2 between 0.47 and 0.61 reported by Xenopoulos and Lodge (2006) for the USA, and may indicate that the SDRs presented here are sufficiently good for use in LCA. Further, our results show that regional SDRs for fish can be calculated for rivers above this latitude, even if the fish diversity is lower due to the postglacial history.

To show the importance of regionally developed Species-discharge relationships we compare our SDRs with the global SDR from Hanafiah et al. (2011) and the Central Plains SDR from Tendall et al. (2014) in Supporting Information 1, Section S5. As our SDRs predict the lowest species richness, our results are in accordance with the statement from Hanafiah et al. (2011) that other existing SDR models should not be applied to rivers north of 42° latitude, due to the low species richness per unit of discharge in these river basins. This highlights that spatially-explicitly developed SDRs are an important requirement (Tendall et al., 2014) to assess the impact of water consumption on a global scale in LCA (Núñez et al., 2016).

To develop the regional SDRs, we identify five regions with similar glacial and dispersal history. In accordance with our assumption that the distance to the refugia is an important factor for recolonization, region 3, located in the southeast of Norway closest to the identified glacial refugia, has the highest species richness. Regions 2 and 5 located in the west of Norway and along the coast, have the lowest species richness, as these regions are further away from the refugia, and could predominantly be colonized by saltwater-tolerant species. However, region 7 located in northern Norway has a higher species richness than

regions 2 and 5, and the same species richness as region 1 located in the most southern part of Norway. This is due to the topography in northern Norway, and indeed in northern Fennoscandia and Russia, which allowed for the postglacial immigration of a diverse fauna of freshwater fish from the east (Huitfeldt-Kaas, 1924).

4.4. Characterization factors

In this study we develop the first CFs quantifying the impact of *net* water consumption on freshwater fish species in Norway, contributing to spatially-explicit regional LCIA models of water consumption impacts on biodiversity. The unit of the CFs is PDF*y/m^3 and is in accordance with existing characterization factors assessing the impacts of water consumption on biodiversity (e.g. Tendall et al., 2014; Verones et al., 2016). In addition, we use the power function as a regression function to ensure comparability which existing characterization factors assessing the impacts of water consumption on biodiversity (e.g. Tendall et al., 2014; Verones et al., 2016). Therefore, this study provides new regional CFs. Novel to this study is that it develops the first method to calculate SDRs in previously glaciated regions. This further indicates that SDRs for northern Europe and northern America can be calculated and used in connection with newly developed CFs. This enables a more regionally specific Life Cycle Impact Assessment, which is needed to assess the biodiversity impact of water consumption on a global scale (Tendall et al., 2014; Núñez et al., 2018).

Hanafiah et al. (2011) report average CFs between $2.51*10^{-15}$ PDF*y/m^3 and $1*10^{-08}$ PDF*y/m^3 below 42° latitude north. Our CFs varying between $7.1*10^{-12}$ PDF*y/m^3 and $8.0*10^{-7}$ PDF*y/m^3 are therefore generally higher. This shows that the impact per fish species of 1 m^3 water consumption in Norway is comparatively higher than that below 42°N. However, as PDFs are calculated relative to the actual species richness in each catchment, only a few potential fish species lost in one catchment could lead to a high PDF value. As our SDRs report a lower fish species richness than the SDRs from Hanafiah et al. (2011), the absolute number of potentially disappeared fish species in Norway could be lower compared to Hanafiah et al. (2011). Our results highlight that spatially-explicit CFs above 42°N are needed to assess the impact of water consumption on a global scale in LCA (Núñez et al., 2016).

However, the SDRs represent a simplification of the relationship between water consumption and biodiversity loss, as frequency and timing of high and low flows, the rate of energy available in a river (Poff and Zimmerman, 2010; Mittelbach et al., 2001), temperature, (Xenopoulos and Lodge, 2006) trophic interactions or habitat diversity also influences fish species richness. Some migratory fish species, for example, require a minimum discharge to migrate (Quinn et al., 1997) and a discharge falling below a certain threshold will lead to a migration stop (Haro et al., 2004). An additional shortcoming of the SDR is that it assumes that fish species cannot rapidly adapt to an altered flow magnitude. Further, the SDR cannot account for external factors like habitat fragmentation (McKay et al., 2013). Our calculated CFs could therefore either lead to an over- or underestimation of the total fish species richness. However, despite all of these shortcomings it has been shown that exactly this simple relationship can be used to identify general patterns between flow and fish species richness (McGarvey, 2014). Therefore, we infer that the SDR can be applied for LCA purposes, as the goal of LCA is to compare general environmental impact patterns between similar products or processes at a global scale (Tendall et al., 2014; Hanafiah et al., 2011; Huijbregts et al., 2016).

The comparison at the global scale further requires CFs with global coverage (Jolliet et al., 2018). Due to its comparably low parameter demand, the SDR enables the development of regionally specific CFs for water consumption impacts on biodiversity at a global scale. However, if appropriate data would be available, the robustness of the SDRs could be greatly increased by including, e.g., species-specific habitat requirements and habitat-discharge interactions (Xenopoulos and Lodge, 2006).

Further, the developed CFs account only for freshwater biodiversity loss due to loss in magnitude of flow, as they are based on the mean annual discharge. As a result, they are not able to assess the effect of seasonality in magnitude change and the related impact on fish species. Our CFs with annual averages thus likely overestimate the impact, as water consumption during a specific season does not necessarily always lead to an impact for all fish species.

4.5. Uncertainty of characterization factors

We use the 95% confidence intervals of the obtained power function coefficients to quantitatively assess uncertainty. In addition, the obtained fish occurrence contributes to the uncertainty of the CFs. However, this uncertainty cannot be assessed quantitatively and therefore is only discussed in a qualitative way in the following section. The obtained fish occurrence data often reflects a strong spatial bias in survey efforts, because some sites are more likely to be surveyed than other sites (Phillips et al., 2009). Also, occurrence data are often collected without planned sampling schemes (Elith et al., 2006). In addition, the probability of detecting a species depends on features of the local habitat or the surrounding landscape (Gu and Swihart, 2004). As a result, the species richness estimation used for the SDR may represent an underestimation. Although not quantifiable, this underestimation is accounted for by weighing the power function by the total number of occurrence records in each catchment (Motulsky and Christopoulos, 2004).

We used all available occurrence points to develop the SDRs, as reservoir operation in Norway began as early 1800. As a result, the developed SDRs may underestimate the fish species richness because we cannot account for fish species that have gotten extinct before the earliest collection date of an occurrence point in the related catchment. The later reservoir operation started in a catchment, the more likely it is that we were able to obtain occurrences points from before reservoir operation. This leads to a lower probability of an underestimation of fish species richness by the SDR. Because the year of reservoir construction varies between catchments, the probability of an underestimation of fish species by the SDR is lowest in region 5 and 7 and highest in region 1 and 3 (SI1, S4).

5. Application in LCA

This study provides *net* water consumption values of Norwegian hydropower reservoirs in combination with CFs quantifying the impact of water consumption on freshwater fish species in Norway. When the *net* water consumption values are implemented in inventory databases and the CFs in Life Cycle Impact Assessment methods, the impact of water consumption of Norwegian hydropower plants on aquatic biodiversity can be assessed on a damage level. When performing an LCA of the whole-life cycle of a storage power plant, water consumption of dam construction and reinvestment phases also have to be considered (Bakken et al., 2016a). Water consumption values for these processes are available in LCI databases (e.g. Wernet et al., 2016). The fact that the CFs vary substantially between the catchments shows that is important to only apply the CF of the relevant catchment in an LCA study and not use average CFs from other catchments, since this may result in a substantial bias in the results. In addition, the CFs in this study should only be used to quantify the impact of a *decrease* in discharge, due to the uncertain influence of *increased* discharge on fish species richness (Xenopoulos and Lodge, 2006).

Finally, we would like to point out that water consumption is only one of several cause-effect pathways leading to potential biodiversity impacts related to hydropower production (Gracey and Verones, 2016), as dam construction for example can also lead to habitat fragmentation (McKay et al., 2013) or influence food web interactions (Power et al., 1996). A holistic LCA of storage power plants should assess all relevant

biodiversity impacts from hydropower electricity production (Gracey and Verones, 2016), thus further model development for the reaming impact pathways is required.

6. Conclusions and future research

This study provides *net* water consumption values of Norwegian hydropower reservoirs in combination with the first developed CFs quantifying the impact of *net* water consumption on freshwater fish species in Norway. Thereby, this study contributes to providing methods and values to assess the biodiversity impact of water consumption. We calculate catchment-specific net water consumption for Norway using reservoir land inundation data in combination with evapotranspiration data. The average net water consumption across all investigated catchments, taking into account evaporation losses *prior* to the inundation of the reservoir, is 0.0016 m^3/kWh. This is 25% smaller than the existing value in the Ecoinvent database (0.002 m^3/kWh) (Flury and Frischknecht, 2012). Further, we develop 1790 catchment-specific characterization factors for Norway, quantifying the aquatic biodiversity impacts of water consumption based on Species-discharge relationships for fish, varying between $7.1*10^{-12}$ PDF*y/m^3 and $8.0*10^{-7}$ PDF*y/m^3. Novel to this CF is that it develops the first method to calculate SDRs in glaciated regions, by delineating regions with similar glacial and fish dispersal history. By using remote sensing assessed reservoir inundated land area (Pekel et al., 2016) and global hydropower reservoirs data (Lehner et al., 2011) in combination with the global MOD16 evaporation model, the methodology for Norway developed in this study has the potential to be applied globally. Further assessment of inundated land area from hydropower reservoirs is thereby most critically needed to allow for the estimation of *net* water consumption values of hydropower reservoirs on a global scale. This study shows that it is possible to calculate regional SDRs and related CFs for fish species in glaciated regions, and therefore additional SDRs for northern Europe and northern America should be calculated and used to develop new CFs. In addition, flow regime alterations have been linked to reduced invertebrate species richness as done by Tendall et al. (Poff and Zimmerman, 2010; Dewson et al., 2007), so developing macro-invertebrate SDRs could be justified in the future. Our CFs developed for Norway can be applied to hydropower projects that aim to include life cycle impacts of existing and planned hydropower reservoirs. Furthermore, a comparison with other energy carriers should be performed, to minimize the highlighted trade-offs between the mentioned SDGs (Nilsson et al., 2016; Bhaduri et al., 2016).

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Appendix A. Supplementary data

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